LCA Case Studies

Application of Life Cycle Assessment to Landfilling

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Abstract

A case study of a life-cycle assessment (LCA) is performed concerning the treatment of household solid wastes in a landfill. The stages considered in this LCA study are: goal and scope definition, inventory analysis and impact assessment. The data of the inventory include the consumption of raw materials and energy through the transport of wastes and the management of landfill, and the corresponding emissions to the environment. Abiotic resource depletion, global warming, acidification, eutrophication and human toxicological impacts have been considered as impact categories for the impact assessment phase of the LCA. A comparison of the environmental impact of the landfilling with and without energy recovery is carried out.

Keywords: Abiotic resources depletion, landfilling; acidification, landfilling; energy recovery, landfilling environmental impact, landfilling eutrophication, landfilling; global warming, landfilling; household solid wastes, treatment; human toxicological impacts, landfilling; landfilling, LCA; LCA; Life Cycle Assessment (LCA); management, landfilling; solid wastes, treatment; wastes, transportation

1 Introduction

The increase of the environmental impact of goods and the reduction of non-renewable material and energy resources, mainly during the last decade, have accelerated the research and development of tools for environmental analysis. One of the most recent of these is the life-cycle assessment (LCA) methodology [1,2]. LCA is divided into four phases: goal and scope definition, inventory analysis, impact assessment and improvement assessment. In the goal and scope definition, the boundaries of the system and the functional unit must be defined. In the inventory phase of the LCA, the inputs and outputs of the system boundaries must be quantified.

In this paper, the LCA methodology is applied to the treatment of household wastes in a landfill in order to evaluate the environmental impact of landfilling through the entire life cycle associated to this activity, consequently allowing the performance of an improvement analysis of this process. In the previous and well-documented work of White et al. [3], a life-cycle inventory of landfilling has been performed using background data. The paper of Finnveden about the emission factors for metals from municipal solid waste landfills [4] must also be mentioned. In the present work, geographical characteristics prevailing in Spain involving an interest for landfill are considered and the possibilities of energy recovery are evaluated.

2 Methodology

2.1 Input analysis

The present LCA is performed by carrying out an inventory of the inputs and outputs related to landfilling. In this inventory, household wastes are considered as an input to the system; in this way, one metric ton of waste is considered to be a functional unit. For the collection of the data, the following steps are taken into account: The transport of wastes from the point of generation to the landfill, the management of the wastes in the landfill, the collection and treatment of the generated biogas and the collection and treatment of the produced leachate. In **Figure 1**, a flow diagram of the system with the corresponding inputs and outputs is presented.

To carry out the inventory, background data has been considered from the bibliography. Quantitative values of resource consumption and emissions are referred to a medium-size landfill that treats 400,000 t annually. On the other hand, climatic data and the chemical composition of household wastes are taken from the mean values registered in Spain.

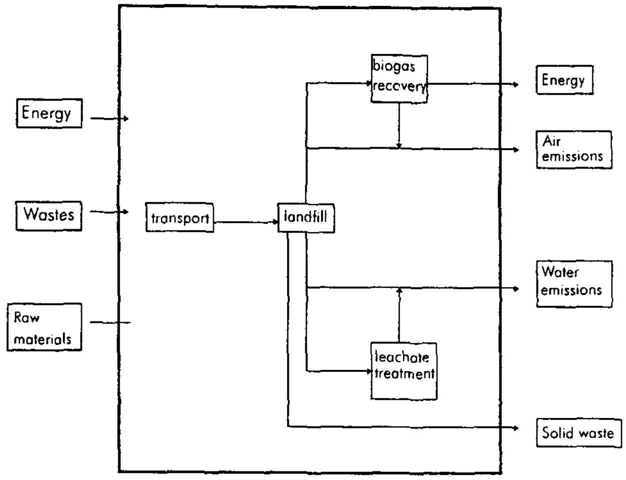

Fig. 1: Flow diagram corresponding to the landfilling process

3 Inventory

Household wastes, raw materials and energy are considered as inputs into the system. In a typical landfill, clay from the surroundings is used as daily covering material. The volume of clay in a landfill is estimated to be 20% of the volume of the deposited wastes [5]. Taking 2.7 and 1.2 t m^{-3} as mean density values for the clay and the wastes in the landfill, respectively [6,7], the amount of clay required per ton of waste is then 0.45 t.

Energy is consumed for the transport of wastes to the landfill and for the management of wastes when deposited. For the calculation of the energy consumed in transport, a mean distance from the point of generation of the wastes to the landfill of 10 km (total: 20 km) is considered. Other parameters that have been considered are: capacity of the truck: 20 m^3, density of wastes inside the truck: 425 kg m^{-3} and diesel consumption: 1l per 2.3 km of distance travelled. With these values and taking into account the energy content of the diesel (37.7 MJ kg^{-1} [8]) and its density (0.9 kg l^{-1} [6]), 34.7 MJ of energy consumptions for the transport of one tonne of waste and the way back of the empty truck (1.73×10^{-3} MJ km^{-1} kg^{-1}) is estimated.

In order to calculate the energy needed for the waste management in the landfill and considering the size of the landfill (capacity of treatment: 400,000 t y^{-1}), two machines working "in situ" are assumed: a bulldozer to move the wastes and a steel wheel compactor to nivellate them, with a diesel consumption of 15 and 18 l h^{-1}, respectively. The time of working has been considered to be 8 h per day and 300 days per year. With a rate of deposition of 400,000 t y^{-1}, an energy consumption of 6.72 MJ per metric ton of waste landfill is deduced. On the other hand, energy is needed for the production of the fuel consumed for the trucks in the transport and the machinery working in the landfill. This energy input has been estimated to be 53.7 MJ per metric ton of waste. For the deduction of this quantity, a value of 44.1 MJ for the extraction, processing and use of 1 l of diesel has been considered [8].

Finally, the feedstock energy in the landfilling process must be considered. This resource input corresponds to the feedstock energy of non-biodegradable material in the wastes that are made from fossil fuels. It is assumed that plastics are the only contributors to the feedstock energy. Considering that

(1) the mean percentage of plastics in Spanish household wastes is 7% [9],
(2) the mean density of plastics is 1 g cm^{-3} [6], and
(3) the mean energy content of plastics is 2 tep m^{-3} [6],

the feedstock energy per metric ton of waste is then estimated to be 5,768 MJ.

3.1 Output analysis

As a consequence of the microbiological processes that occur in the landfill, biogas is produced. A production of biogas of 200 m^3 per metric ton of waste with a gas collection efficiency of 55% and a calorific value of the biogas of 19.5 MJ m^{-3} is assumed [7]. If the biogas collected is flared to recover energy, an energy generation of 2,145 MJ per metric ton of waste can be deduced.

Other outputs from the system to the environment are emissions to the atmosphere and hydrosphere, and solid waste emissions. **Table 1** summarizes the emissions to the atmosphere due to transport and management of wastes, considering a respective consumption of 1.0 l and 0.2 l of diesel per metric ton of waste for these activities. These values are deduced from data corresponding to the emissions to the atmosphere due to the production and use of diesel [8].

Table 1: Emissions to the atmosphere in g per metric ton of wastes landfill due to the transport and management

emissions	transport	management
CO	26.5	5.3
CO_2	3036.3	607.2
NO_x	33.9	6.8
SO_x	10.1	2.0
C_xH_y	10.9	2.2

Other emissions to the atmosphere correspond to those produced by biogas. It is considered a mean composition of 55% CH_4 and 45% CO_2, with the remaining 1% mainly being hydrocarbons [10]. Taking into account the biogas production in the landfill and the collection efficiency previously mentioned, the emissions to the atmosphere per metric ton of waste are: 34.480 g of CH_4 and 292.431 g of CO_2. In the calculations, it has been assumed that all CH_4 is converted stoichiometrically to CO_2 during the flaring of the collected biogas. In these estimates, the minor gases in the biogas and those formed during combustion (e.g., NO_x, SO_x, dioxines, etc. ...) have been neglected. If energy is not recovered and biogas escapes to the atmosphere, then the emissions of CH_4 and CO_2 become 76,622 g and 176,540 g, respectively. In **Table 2**, the total emissions to the atmosphere for the landfilling of one metric ton of waste with and without energy recovery are summarized.

Table 2: Total emissions to the atmosphere in g per metric ton of wastes landfill with and without energy recovery

emissions	without energy recovery	with energy recovery
CO	31.8	31.8
CO_2	180183.4	296074.4
CH_4	76622	34480
NO_x	40.7	40.7
SO_x	12.1	12.1
C_xH_y	13.1	13.1

The emissions to the hydrosphere are related with the generation of leachates and with the emissions from transport and management of the wastes. To estimate the production of leachate of the wastes in the landfill during its life, the following parameters have been taken into account:

(1) water content in the wastes: 55% [11],
(2) average annual rainfall: 600 mm,
(3) average depth of landfilled wastes: 20 m,
(4) field capacity of wastes and clay in the landfill: 30% [12],
(5) mean density of wastes in the landfill: 1.2 t m^{-3} being [7] and
(6) period of leachate production until landfill closure: 30 years.

To estimate the leachate originated from the rain, it is assumed that 35% is lost by evapotranspiration, 20% is lost by surface runoff and 30% is retained in the landfill [12]. Consequently, 15% of rain emerges as leachate; this implies that 112.5 l of leachate per metric ton of waste is produced from rain.

Another contribution to the production of leachate comes from the water content of the wastes. In this way, considering that 30% of water is retained by the wastes, the remaining 25% emerges as leachate. Consequently, the volume of water per metric ton of wastes from this contribution is 250 l. The density of water has been taken as being equal to 1 g l^{-1}. Finally, the water which reacted with the organic matter through the fermentation process must be taken into account. Assuming a mean elemental composition of the organic fermentable waste of 68:111:50:1 for C:H:O:N (13), the corresponding anaerobic process can be written as:

$$C_{68}H_{111}O_{50}N + 16H_2O \rightarrow 35CH_4 + 33CO_2 + NH_3$$

Considering that household wastes contain 70% of organic fermentable matter (14), a water consumption of 52.1 l per metric ton of waste through the above chemical reaction can be deduced. As a consequence, the total production of leachate per metric ton of waste during its active life is 310.4 l.

To calculate the emissions to the hydrosphere of the different chemical species, a mean chemical composition of the leachates produced in a landfill considered intermediate between a young and a mature one has been taken into account [15]. On the other hand, it has been considered that only the 70% of the leachate is collected, with the remaining 30% being leaked out [4]. It is assumed that the collected leachate is treated in a sewage treatment plant in which 90% of the COD and DBO is eliminated as well as 100% of the ammoniac, the organic nitrogen and the metallic cations, except the alkaline ions [16].

The emissions to the hydrosphere coming from transport and waste management are associated with the extraction and processing of the fuel [4]. These emissions are considered negligible because they amount to s than 1% of the corresponding leachate emissions, except for particulates and metals where emissions of 14.5 g per metric ton of waste have been estimated for both substances [4]. Table 3 summarizes the emissions to the hydrosphere per metric ton of waste landfill.

Table 3: Emissions to the hydrosphere in g per metric ton of wastes landfill

emissions	mass
COD	1436.0
BOD	689.3
SS	42.4
$N\text{-}NH_3$	56.0
N-org	0.9
NO_3^-	0.5
PO_4^{3-}	0.3
Cl^-	512.0
Na^+	350.5
K^+	212.6
Me	14.5

ss: solids in suspension
Me: metals

To estimate the solid waste emissions, a mass-balance must be performed. From the initial 0.45 t of deposited dry waste, 0.315 t have reacted in the anaerobic process. Considering that 0.45 t of clay is used to cover one metric ton of deposited waste, the remaining inert solid in the landfill is 0.585 t. On the other hand, from the initial 0.55 t of water contained in one metric ton of waste, 0.25 t have leached and 0.052 t have reacted with organic material. To complete this mass balance, it must be considered that 0.585 t of dry solid retain 0.175 t of water (field capacity: 30%), while the remaining water evaporates due to the high temperatures inside the landfill [17]. As a consequence, the quantity of waste remaining in the landfill is 0.76 t, which corresponds to a mass reduction of 24% of the initial waste deposited in the landfill.

4 Impact Assessment

To carry out this phase of the LCA, the following impact categories have been considered: global warming (GW), acidification (A), eutrophication (E) and human toxicology (HT).

Table 4: Classification of the emissions into the following impact categories: Global Warming (GW), Acidification (A), Eutrophication (E) and Human Toxicology (HT)

GW	A	E	HT
CO_2	NO_x	NO_x	NO_x
CH_4	SO_x	$N\text{-}NH_3$	SO_x
		N-org	CO
		NO_3^-	NO_3^-
		PO_4^{3-}	$C_xH_{y(g)}$
		DQO	$Me_{(aq)}$

In Table 4, the inputs and outputs in the inventory are classified in different impact categories. To quantify the potential contribution of an input or output j to the impact i (P_{ij}), the following relation is applied:

$$P_{ij} = A_j W_{ij}$$

where A_j is the amount of input or output and W_{ij} is the weighting factor. The total potential contribution to the effect from all inputs and outputs (P_i) is calculated from:

$$P_i = \Sigma_j P_{ij} = \Sigma_j A_j W_{ij}$$

The weighting factors used to quantify the environmental impacts are taken from the bibliography [18] and are summarized in **Table 5**. Mean values for the human toxicology impact weighting factors of $C_xH_{y(g)}$ and metals have been taken. Using these weighting factors and the mass of the emissions, the corresponding potential contributions to the different impact categories have been determined. These values are showed in **Table 6**, in which the three stages: transport, management and biodegradation of wastes in the landfill with and without energy recovery are distinguished.

Table 5: Weighting factors used to quantify the environmental impact [18]

contaminant	GW	A	E	HT
CO_2	1			
CH_4	62ᵃ			
NO_x		0.7	0.13	0.78
SO_x		1		1.2
$N\text{-}NH_3$			0.33	
N-org			0.42	
NO_3^-			0.1	7.8×10^{-4}
PO_4^{3-}			1	
DQO			0.022	
CO				1.2×10^{-2}
$C_xH_{y(g)}$				1
$Me_{(aq)}$				10^{-2}

ᵃtime frame: 20 years

As can be seen, biodegradation is the stage of landfilling that contributes more to global warming and eutrophication, while transport is the main contributor to acidification and human toxicology impacts. It is worth noting the beneficial environmental effect of energy since it reduces the global warming potential of the landfilling by about 50%. Finally, abiotic resource depletion (ARD) can also be considered as an impact category. In this way, the contribution of this impact category comes from the feedstock energy (5,768 MJ – see the inventory section) and from clay consumption (0.45 t per metric ton of waste – see the inventory section). The last stage of the LCA is the valuation in which the relative importance of the different environmental impacts are weighted against each other. The stage is based mainly on political, ethical and administrative considerations and lies beyond the scope of this study.

Table 6: Potential contributions of the emissions to the different impact categories for the three relevant stages of landfilling: transport, management and biodegradation

stage	GW/Kg		A/g	E/g	HT/g
	with energy recovery	without energy recovery			
transport	3.0	3.0	33.8	4.4	49.8
management	0.6	0.6	6.8	0.9	10.0
biodegradation	2430	4927		50.8	0.14

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